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Potential impacts of the invasive herb garlic mustard (*Alliaria petiolata*) on local ant (Hymenoptera: Formicidae) communities in northern temperate forests.

KALOYAN IVANOV^{1,*} AND JOE KEIPER²

ABSTRACT

Garlic mustard, an invasive shade-tolerant species introduced to North America from Eurasia in the late 1860s, now is widely distributed throughout the US and Canada. The presence of garlic mustard results in displacement of understory species and subsequent decline in native plant diversity. By displacing native plants, garlic mustard can affect resource availability and habitat quality, and thereby affect animals across different trophic levels. However, these impacts have been documented infrequently. Our study focused on the small-scale effects of garlic mustard invasion on an abundant and important group of forest-floor arthropods. We evaluated the effects of garlic mustard on forest ant assemblages in invaded and non-invaded areas of two mesophytic forest fragments of northeastern Ohio. Plots invaded by garlic mustard showed reduced leaf litter depth, and an increased abundance of nonnative *Amyntas* earthworms. Sample-based rarefaction and similarity analyses revealed that the presence of garlic mustard, and the associated decrease in leaf litter depth, had no detectable effect on the observed and expected ant species richness and community composition. Rank-abundance distributions also were largely unchanged in garlic mustard invaded plots. Our results suggest that regional sylvan ant communities are unaffected by the generally presumed negative effects of garlic mustard invasions, or these effects may be more subtle or confounded by other dominating factors.

Keywords: Ohio, species richness, abundance, community composition, leaf-litter, exotic earthworms

INTRODUCTION

Invasion of alien plant species is considered among the most significant threats to biodiversity (Blossey 1999). Non-native species can displace native plants, change trophic structure and alter ecosystem processes (Mack et al. 2000; Ehrenfeld 2003). Problems associated with invasive plant species have increased greatly in recent times due to increased species exchange between geographic areas, largely as a result of human-aided dispersal (Vitousek et al. 1997; Chapin et al. 2000). In

addition, human-mediated habitat disturbances can accelerate the incidence of successful biological invasions by enhancing the establishment of exotic species (Liebhold et al. 1995; King and Tschinkel 2006). Despite the increased awareness of the negative impacts of invasive plants, there is little quantitative evidence for the ecological impacts they pose (Blossey 1999). The widespread invasive garlic mustard (*Alliaria petiolata* [Bieb.] Cavara and Grande) is no exception, and its ecological impacts though commonly suspected have been poorly documented (Blossey et al. 2001; Dávalos

*Corresponding author K. Ivanov

1. Cleveland Museum of Natural History, 1 Wade Oval Drive, University Circle, Cleveland, OH 44106. e-mail: antzmail@gmail.com. telephone: 216-231-4600 ext. 3314. fax: 216-231-5919

2. Virginia Museum of Natural History, 21 Starling Avenue, Martinsville, VA 24112

and Blossey 2004).

Garlic mustard (Brassicaceae) is a shade-tolerant, short-lived, obligate biennial herb native to western Eurasia (Welk et al. 2002). It was introduced to North America from Europe in 1868 when it was first recorded in Long Island, NY (Nuzzo 1993), and has quickly spread throughout the US and Canada. It is most abundant in the northeastern and midwestern US (Welk et al. 2002; Blossey et al. 2002). Garlic mustard produces first year rosettes that over-winter, it flowers early in the following year, and plants die after seed production. A single plant can produce hundreds of seeds, with maximum yields of 7900 seeds (Nuzzo 1993). Garlic mustard spreads exclusively by seeds, which require a dormancy period of 8 to 22 months to germinate and can remain viable for up to five years (Nuzzo 1999; Blossey et al. 2002). *Alliaria petiolata* typically occupies disturbed habitats and often is most abundant in urban forest fragments and forest edges, although it has the unusual capacity to invade and proliferate within intact forest communities (Nuzzo 1999; Dávalos and Blossey 2004; Rodgers et al. 2008).

Garlic mustard is one of the most problematic invaders in North American forest communities where it becomes a permanent member upon introduction (Rodgers et al. 2008). Among the most often reported negative impacts are the displacement of native understory plants and declines in the diversity of native plant communities (McCarthy 1997; Meekins and McCarthy 1999; Stinson et al. 2007), although quantitative evidence often is inconclusive (Blossey et al. 2001). Garlic mustard decreases the abundance of mycorrhizal fungi in the soil and on plant roots (Stinson et al. 2006), which may reduce the competitive ability of many native plant species (Roberts and Anderson 2001). The disruption of mycorrhizal associations within understory plant communities has been related to the production of secondary compounds by *A. petiolata* used to deter herbivory and to suppress the growth of plant and fungal species (Fahey et al. 2001; Cipollini and Gruner 2007; Rodgers et al. 2008).

Given garlic mustard's impacts on local

forest flora, arthropod diversity may decline, as it commonly is thought to be positively correlated with floristic and structural plant diversity (Siemann et al. 1998). Arthropod herbivore richness is generally greatest in areas of highest plant richness, primarily as a result of greater availability of resources (Siemann 1998; Knops et al. 1999). This in turn results in a greater diversity of arthropod predators and parasitoids (Hunter and Price 1992; Siemann 1998). Increased plant diversity also may directly increase diversity of higher trophic levels through an increase in floral resources (Jervis et al. 1993). Plant invasions and the associated decrease in plant diversity and homogenization of local flora can alter invertebrate assemblages. Changes in invertebrate abundance, richness and community composition have been reported for a variety of arthropod groups in areas invaded by non-native plants (Standish 2004; Gratton and Denno 2005; Wilkie et al. 2007; Bultman and DeWitt 2008).

The presence of garlic mustard has been shown to interfere with the development of the native butterflies *Pieris napi oleracea* Harris and *P. virginensis* Edwards (Lepidoptera: Pieridae). Females of these species oviposit on garlic mustard plants, instead on their native toothwort hosts, but garlic mustard increases the time required for larval development, larvae take longer to pupate, and exhibit reduced pupal mass and pupation rates (Porter 1994; Huang et al. 1995; Keeler and Chew 2008). Dávalos and Blossey (2004) reported no effect of garlic mustard presence on ground carabid beetle captures and species richness, and on overall invertebrate abundance in New York deciduous forests. However, the authors were able to document a significant decrease in leaf litter depth at garlic mustard invaded plots which was associated with the presence of non-native earthworms. A positive correlation between garlic mustard cover and non-native earthworm biomass, and a negative correlation with leaf litter depth also have been reported by Nuzzo et al. (2009) in New York and Pennsylvania. Garlic mustard invasions therefore also may be symptomatic of invasions by non-native earthworms.

The displacement of native understory plant

species associated with the presence of garlic mustard and the creation of near-monospecific stands in the areas where this species occurs are likely to alter resource availability, habitat quality and microclimate and thus affect organisms from different trophic levels. In addition, areas invaded by garlic mustard may suffer reduction in the leaf litter layer as a direct result from the activity of non-native earthworms. This is likely to negatively affect forest dwelling invertebrates that rely on leaf litter as nesting and foraging resource.

Our research focused on the small-scale effects of garlic mustard invasions on ground dwelling forest ant communities. Ants are important and nearly ubiquitous members of terrestrial ecosystems. They play major ecological roles as predators, scavengers, mutualists, seed gatherers and dispersers (Hölldobler and Wilson 1990). Ants participate in a variety of mutualistic relationships with plants, fungi and animals, and also are an important prey item for both vertebrates and invertebrates (Hölldobler and Wilson 1990; Folgarait 1998). Their ease of sampling, stationary colonies, and responsiveness to environmental disturbances have made them an attractive monitoring tool for tracking changes in ecosystem conditions (Andersen 1990, 1993, 1997; Alonso 2000; Kaspari and Majer 2000). The changes initiated by garlic mustard invasions could affect local ant communities both directly and indirectly and result in changes in local species richness, community composition and structure. Direct impacts can result from changes in habitat quality and resource availability associated with the presence of garlic mustard. In addition, the changes in local plant diversity may give rise to indirect impacts through disruption of mutualistic interactions between ants and honeydew-producing 'Homoptera' and/or between ants and ant-dispersed (myrmecochorous) plants. Although not obligately dependent on their mutualistic partners, the food resources in the form of honeydew or elaiosomes are an important part of the diet of many ant species in temperate deciduous forests. Changes in the species richness and composition of the local ant communities may lead to modifications of species interactions (competition, predation, parasitism,

pollination, and seed dispersal), and alteration of ecosystem processes (Hölldobler and Wilson 1990; Murcia 1995; Folgarait 1998; Laurence et al. 2002).

To our knowledge, no study has explored the impacts of garlic mustard invasions on ant communities. Here we report the results of a study designed to evaluate the potential impacts of garlic mustard invasions on local temperate-forest ant communities. We investigated whether ant communities differ in abundance, species richness, and composition between sites invaded or not invaded by garlic mustard. We expected lower species richness and abundance of ants in areas invaded by garlic mustard as a result of alterations in habitat quality, resource availability, and/or disruption of important mutualistic interactions.

MATERIALS AND METHODS

STUDY SITES

We established two study sites in temperate mixed deciduous forest fragments part of the Cleveland Metroparks in the Cleveland area of northeastern Ohio. One site was located at Bedford Reservation (BED; 890 ha; 41° 22' N, 81° 33' W), and a second one at Brecksville Reservation (BRE; 1406 ha; 41° 18' N, 81° 36' W). At each site we established 10 plots, of which five were invaded by garlic mustard and five were reference plots. Invaded plots were selected on the basis of visual estimates of the presence of the target species in 30% or more of the cover in the plots. Garlic mustard was present at all of the invaded plots and in none of the non-invaded plots. A few garlic mustard rosettes were present near the edges of some reference plots, but none occurred within the plots. Each plot covered an area of 25m x 25m, and plots within sites were separated by a minimum distance of approximately 150m. The elevation of all plots ranged between 200 and 285m a.s.l.

Both research sites were located within mixed deciduous forest types, dominated by mixtures of red and white oak (*Quercus rubra* L. and *Q. alba* L.), sugar and red maple (*Acer saccharum* Marshall and

A. rubrum L.), American beech (*Fagus grandifolia* Ehrh.), tulip tree (*Liriodendron tulipifera* L.), and hickories (*Carya* spp.), supplemented to varying degrees by black cherry (*Prunus serotina* Ehrh.), cucumber tree (*Magnolia accuminata* (L.)), and basswood (*Tilia americana* L.). The understory is characterized by seedlings and saplings of the canopy trees; the shrubs *Lindera benzoin* L., *Hammamelis virginiana* L., and *Viburnum* spp.; the vines *Toxicodendron radicans* (L.), *Parthenocissus quinquefolia* (L.); and various herbs and graminoids. Plant nomenclature follows Gleason and Cronquist (1991).

SAMPLE COLLECTING AND PROCESSING

Ants were collected via Winkler litter extraction in the period June-July during the peak of annual ant activity in the region. Litter collection was conducted between 10.00 and 16.00h, and at least a day after a heavy rain as ants are extracted less effectively from wet litter (Fisher 1998).

Five 1m² quadrats were placed in a dice pattern within each plot, with the center of the dice corresponding to the center of the research plot. The closest quadrats (diagonal from center to rest of quadrats) were separated by 7m, and all quadrats were more than 6m away from the edges of the plot to increase the likelihood that foragers originated from within the study plot. We collected the leaf litter and the top layer of loose soil within each quadrat and sifted the collected material through a sifter with a mesh opening of approximately 10mm to exclude larger leaves, twigs and stones. In the laboratory, we loaded the sifted litter into Winkler extractors and left them to operate at room temperature for 72h. Three days are sufficient to extract nearly all ant species and the majority of individuals from the collected samples (Ivanov et al. 2010). At the end of the extraction time, we rinsed the content of each collecting container into a labeled vial containing 95% ethanol.

We sorted, counted and identified all individuals to species using the taxonomic keys in Smith (1957), Coover (2005), and Francoeur (2007). We consulted the ongoing work of A. Francoeur, who is revising the North American *Myrmica*, to account

for the presence of a yet undescribed species in our samples. Vouchers will be deposited at the Cleveland Museum of Natural History, Department of Invertebrate Zoology, and the remaining materials are in the first author's collection. Ant nomenclature follows Bolton et al. (2007).

We measured garlic mustard density by counting the number of stems present in two 0.25m² quadrats placed within each invaded plot. These quadrats were positioned 5m away, on each side, from the central ant-sampling quadrat. In addition, we recorded the depth of the litter layer, to the nearest mm, at the four corners and the center of each sampling quadrat and averaged the five measurements for a single value for each sampling quadrat. In cases of large discrepancies between the individual measurements (>25%), caused by uneven litter accumulation or unevenness in the surface profile of the underlying soil, we discarded the lowest and highest values and took two additional measurements.

At the onset of our sampling at BED, we observed high numbers of earthworms present in the leaf litter layer of the garlic mustard invaded plots and we decided to include an estimate of the density of the earthworms present at the research plots in our study. As we did not record the earthworm abundance in the initial plots sampled we limited our earthworm counts to the plots located at BRE. We counted all earthworms present in the leaf-litter layer and on the soil surface within each of the five one square meter samples, and identified representative specimens using the keys in Hale (2009). We did not use earthworm-specific extraction techniques, such as application of mustard solution or formalin, as temperate *Amyntas* typically are epigeic (residing in the leaf litter layer), or epi-endogeic (surface soil; Hendrix and Bohlen 2002). Our approach therefore allows for a valid estimation of only the surface density of *Amyntas* earthworms at the research plots.

DATA ANALYSIS

We compared *Alliaria* density between BED and BRE invaded plots using a two-sample t-test; leaf-litter depth using a two-way Analysis of Variance (ANOVA), with garlic mustard presence and site

as main factors; and *Amyntas* density between BRE invaded and reference plots using the non-parametric Mann-Whitney U test. Normality and homoscedasticity of the data were evaluated prior to the analyses with a Kolmogorov-Smirnov (K-S) test and Levene's test, respectively.

We used a two-way ANOVA to compare ant abundance and species density, with garlic mustard presence and site identity as main factors. We used sample-based rarefaction to calculate and compare the observed and expected species richness across the pooled invaded and non-invaded plots, and across sites (rarefaction curves were scaled to number of occurrences; Gotelli and Colwell 2001). Using incidence and not abundance data is necessary as the sociality of ants often leads to clumping of individuals within samples that can skew species-richness comparisons and species-abundance relationships (King and Porter 2005). We created all sample-based rarefaction curves using the analytical method of Colwell et al. (2004), implemented in EstimateS 8.2 (Colwell 2009). To estimate the asymptotic (total) species richness we calculated the Chao2 (Chao 1987) estimator, using 100 randomizations of sample accumulation order. This estimator relies only on presence/absence data, and has been found to be one of the least biased and the most precise estimation methods (Colwell and Codington 1994; Walter and Moore 2005; Hortal et al. 2006). For the analyses, data were pooled for all replicate samples within the invaded and non-invaded plots and compared using the 95% confidence intervals provided in the EstimateS output. In addition, we used the calculated measures of expected species richness to assess the degree of sampling completeness in our study.

We used hierarchical clustering (group-average linking algorithm with Bray-Curtis similarity measure) to identify natural groupings among the study plots according to similarities in their ant species composition. We used a nonparametric two-way Analysis of Similarity (ANOSIM, Clarke 1993), with garlic mustard and site as main factors, to test for differences in species composition between the invaded and non-invaded plots. Both analyses were performed with the software package

PAST (version 1.97, Hammer et al. 2001).

We constructed rank-abundance plots and compared them with the nonparametric Kolmogorov-Smirnov test to assess any differences in the structure of the ant communities between the invaded and non-invaded plots (Gotelli and Ellison 2004; Ellison et al. 2007). We used SPSS 16.0 (SPSS Inc, Chicago, Illinois) for all statistical analyses, unless otherwise noted in the text.

RESULTS

The average number of garlic mustard stems per quadrat was similar in the invaded plots at the two study sites ($98.7 \pm 14.6\text{SE}$ at BRE, and $146.3 \pm 29.3\text{SE}$ at BED; $t = 1.45$; $df = 18$; $P = 0.16$).

We collected a total of 9922 ant workers representing 25 species from 12 genera (Table 1). Of these, 4470 ants were collected at BRE and 5452 at BED, with 4639 ants collected at the reference plots, and 5283 at the invaded plots. Ant abundance per sample was higher for the non-invaded plots at BED, but exhibited the opposite trend at BRE, being higher at the invaded plots (Fig. 1). A two-factor analysis of variance showed no significant

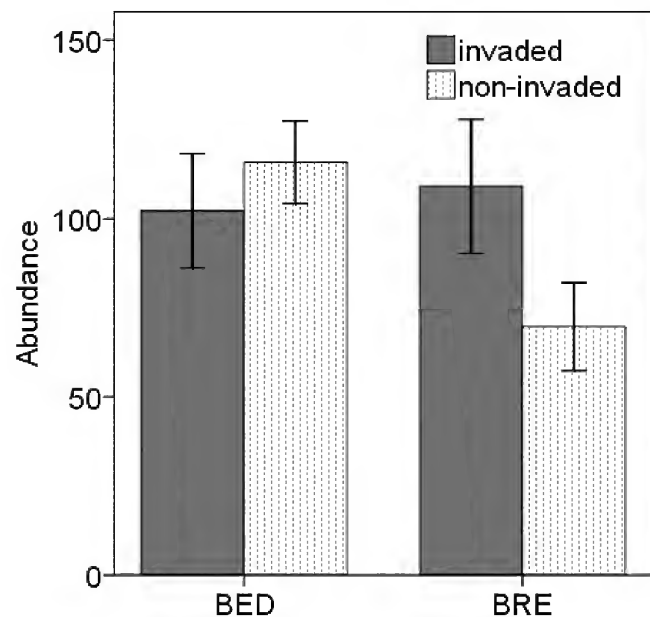


Fig. 1 Average ant abundance at garlic mustard invaded and non-invaded plots. Error bars are $\pm 1\text{SE}$.

Table 1. Ant species, with corresponding abundances, collected in garlic mustard invaded and non-invaded plots at Bedford and Brecksville.

| Species | BEDFORD | | BRECKSVILLE | | TOTAL | |
|--|-------------|---------|-------------|---------|-------------|---------|
| | non-invaded | invaded | non-invaded | invaded | non-invaded | invaded |
| Amblyoponinae | | | | | | |
| <i>Amblyopone pallipes</i> (Haldeman, 1844) | 44 | 12 | 15 | 13 | 59 | 25 |
| Ponerinae | | | | | | |
| <i>Ponera pennsylvanica</i> Buckley, 1866 | 132 | 290 | 66 | 21 | 198 | 311 |
| Myrmicinae | | | | | | |
| <i>Myrmica punctiventris</i> Roger, 1863 | 389 | 132 | 509 | 270 | 898 | 402 |
| <i>Myrmica semiparasitica</i> Francoeur, 2007 | 3 | 0 | 0 | 11 | 3 | 11 |
| <i>Myrmica</i> sp. | 0 | 19 | 0 | 0 | 0 | 19 |
| <i>Stenamma brevicorne</i> (Mayr, 1886) | 154 | 232 | 6 | 32 | 160 | 264 |
| <i>Stenamma impar</i> Forel, 1901 | 124 | 152 | 10 | 76 | 134 | 228 |
| <i>Stenamma schmitti</i> W.M. Wheeler, 1903 | 142 | 13 | 1 | 9 | 143 | 22 |
| <i>Aphaenogaster picea</i> (W.M. Wheeler, 1908) | 387 | 457 | 301 | 859 | 688 | 1316 |
| <i>Aphaenogaster tennesseensis</i> (Mayr, 1862) | 0 | 1 | 0 | 0 | 0 | 1 |
| <i>Temnothorax curvispinosus</i> (Mayr, 1866) | 177 | 53 | 141 | 42 | 318 | 95 |
| <i>Temnothorax longispinosus</i> (Roger, 1863) | 1 | 6 | 3 | 2 | 4 | 8 |
| <i>Temnothorax schaumii</i> (Roger, 1863) | 0 | 1 | 0 | 1 | 0 | 2 |
| <i>Myrmecina americana</i> Emery, 1895 | 663 | 489 | 150 | 575 | 813 | 1064 |
| Dolichoderinae | | | | | | |
| <i>Tapinoma sessile</i> (Say, 1836) | 12 | 1 | 4 | 0 | 16 | 1 |
| Formicinae | | | | | | |
| <i>Prenolepis imparis</i> (Say, 1836) | 18 | 47 | 0 | 4 | 18 | 51 |
| <i>Lasius alienus</i> (Foerster, 1850) | 590 | 605 | 347 | 795 | 937 | 1400 |
| <i>Lasius nearcticus</i> W.M. Wheeler, 1906 | 0 | 0 | 0 | 1 | 0 | 1 |
| <i>Lasius umbratus</i> (Nylander, 1846) | 0 | 19 | 33 | 0 | 33 | 19 |
| <i>Formica neogagates</i> Emery, 1893 | 0 | 0 | 98 | 0 | 98 | 0 |
| <i>Formica subsericea</i> Say, 1836 | 33 | 0 | 45 | 10 | 78 | 10 |
| <i>Camponotus chromaiodes</i> Bolton, 1995 | 3 | 7 | 12 | 1 | 15 | 8 |
| <i>Camponotus pennsylvanicus</i> (De Geer, 1773) | 14 | 16 | 1 | 4 | 15 | 20 |
| <i>Camponotus nearcticus</i> Emery, 1893 | 6 | 1 | 1 | 1 | 7 | 2 |
| <i>Camponotus subbarbatus</i> Emery, 1893 | 4 | 3 | 0 | 0 | 4 | 3 |

effect of garlic mustard presence ($F_{(1,96)} = 0.74$, $P = 0.39$), and site identity ($F_{(1,96)} = 1.72$, $P = 0.19$) on ant abundance, with no interaction between the main factors ($F_{(1,96)} = 3.13$, $P = 0.08$; Fig. 1). Similar results were obtained when abundances were compared at the plot, rather than the sample level.

At BRE, we collected a total of 22 species of which 18 were recorded from the non-invaded and 19 species from the invaded plots, with 15 species being common to both plots. At BED, we recorded a similar number of species (23), of which 19 were present at the non-invaded plots, and 21 at the invaded plots. Seventeen species were common to both plots at BED (Table 1). The most abundant species in our samples were *Lasius alienus* (Foerster, 1850), *Aphaenogaster picea* (W.M. Wheeler, 1908), *Myrmecina americana* Emery, 1895, and *Myrmica punctiventris* Roger, 1863 comprising 68% and 85% of the total captures at BED and BRE, respectively (Table 1). No non-native ant species occurred in the collected samples. Both observed and expected species richness were higher at the invaded plots, but the differences were not significant. Our rarefaction analyses showed no significant difference in observed species richness between the invaded and non-invaded plots (Fig. 2). In addition, there was no difference in observed species richness between the invaded and non-invaded plots when compared across sites. The expected number of species was estimated at 28.7 and 21.7 species for the invaded

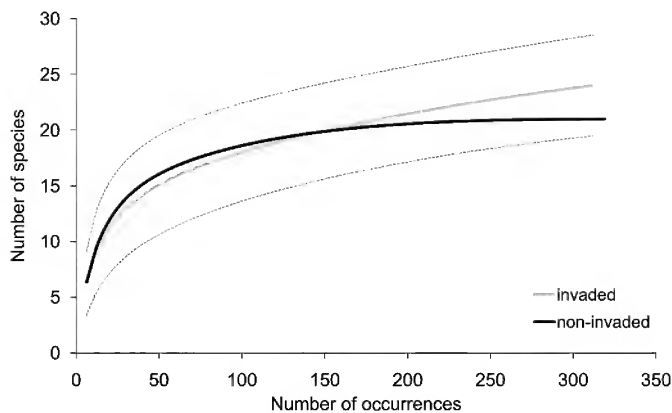


Fig. 2 Rarefaction curves for pooled invaded and non-invaded plots. For clarity the 95% confidence intervals (lighter dotted lines) are shown only for the invaded plots.

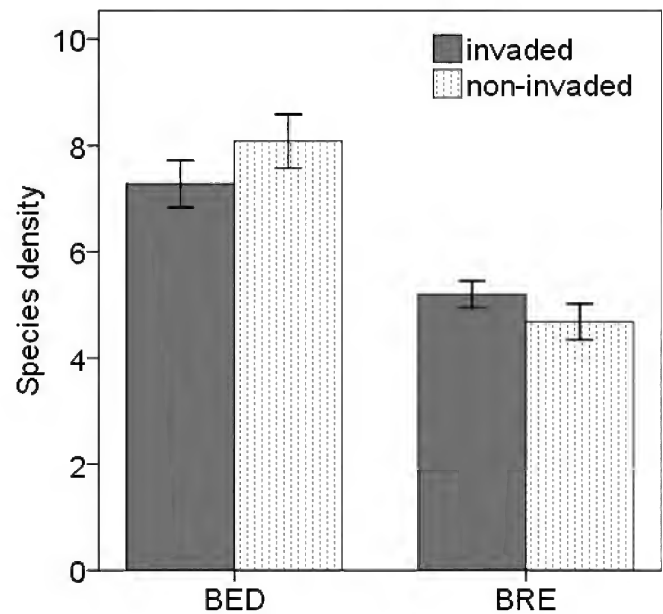


Fig. 3 Species density at garlic mustard invaded and non-invaded plots. Error bars are ± 1 SE.

and non-invaded plots, respectively. The difference was not significant as evidenced by the overlap of the 95% confidence intervals. Based on these estimates, we were able to sample an estimated 84% and 97% of the local ant fauna in invaded and non-invaded plots, respectively. Small scale species richness (i.e., species density per sample) was higher at BED than at BRE (two-way ANOVA: $F_{(1,96)} = 47.88$, $P < 0.001$), but not different across the invaded and reference plots within a site ($F_{(1,96)} = 0.13$, $P = 0.72$), with no interaction between the main factors ($F_{(1,96)} = 2.78$, $P = 0.10$; Fig. 3).

The captures of some species differed between the non-invaded and the invaded plots (Table 1), however these patterns were often inconsistent across sites. For example, *Myrmica semiparasitica* Francoeur, 2007 was found only at the reference plots at BED, but was found only at the invaded plots at BRE. A reversed pattern was observed for *Lasius umbratus* (Nylander, 1846). Four species occurred only at the invaded plots, but three of those were restricted to only one of the two study sites. All four of these species showed very low frequency of occurrence, being present in only one or two samples (Table 1). *Formica neogagates* Emery, 1893 was the only species that occurred at just non-invaded plots, however it was only found

at BRE and represented only 1% of the total of all ants collected.

Our hierarchical cluster analysis did not reveal clear separation of plots, based on either presence of garlic mustard or site (Fig. 4). A two-way ANOSIM revealed no significant difference in ant species composition between invaded and non-invaded plots ($R = 0.006$, $P = 0.47$) and between sites ($R = 0.056$, $P = 0.26$). Our analyses thus show no evidence for differences in local ant community composition between sites invaded and not invaded by garlic mustard. In addition, we found no changes in the structure of the local ant communities between the invaded and non-invaded plots (Kolmogorov-Smirnov two-sample test: $Z = 0.707$, $P = 0.70$; Fig. 5).

Leaf-litter depth was significantly lower in the invaded plots at both sites (two-way ANOVA: $F_{(1,96)} = 40.76$, $P < 0.001$), with no significant effect of site ($F_{(1,96)} = 1.32$, $P = 0.25$), and no interaction between the main factors ($F_{(1,96)} = 2.81$, $P = 0.10$; Fig. 6). At BRE, we found a significantly higher number of *Amyntas* earthworms present at the invaded plots ($64.4 \pm 20.6\text{SD}$; Mann Whitney U test: $Z = -5.63$, $P < 0.001$), as compared to the non-invaded plots ($0.6 \pm 1.7\text{SD}$).

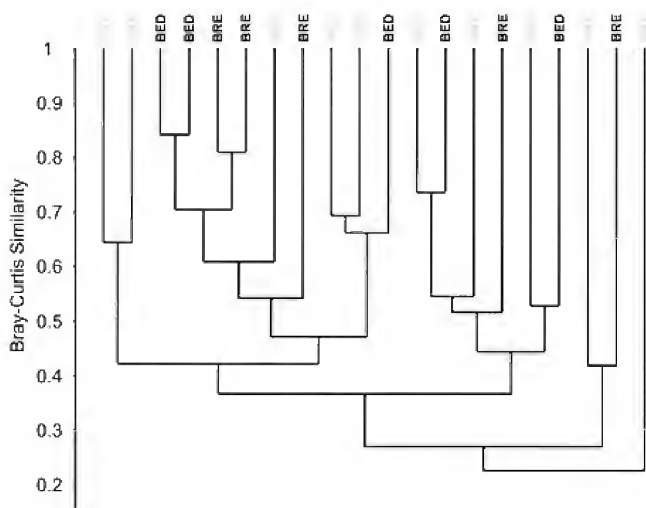


Fig. 4 Dendrogram for hierarchical clustering of invaded and non-invaded plots according to similarities in ant species composition. Invaded plots are presented in black bold type, and reference plots are shown in grey

DISCUSSION

At the local scale at which we conducted our study, and contrary to our expectations, we were unable to detect impacts of garlic mustard presence on native forest ant assemblages. The presence of garlic mustard was not associated with detectable changes in the ant communities at the study sites, and our results showed no significant differences in terms of abundance, species richness or composition

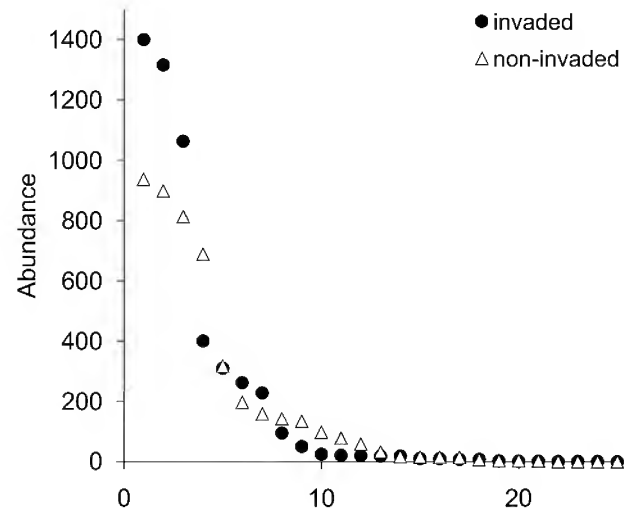


Fig. 5 Rank abundance plots for garlic mustard invaded and non-invaded plots

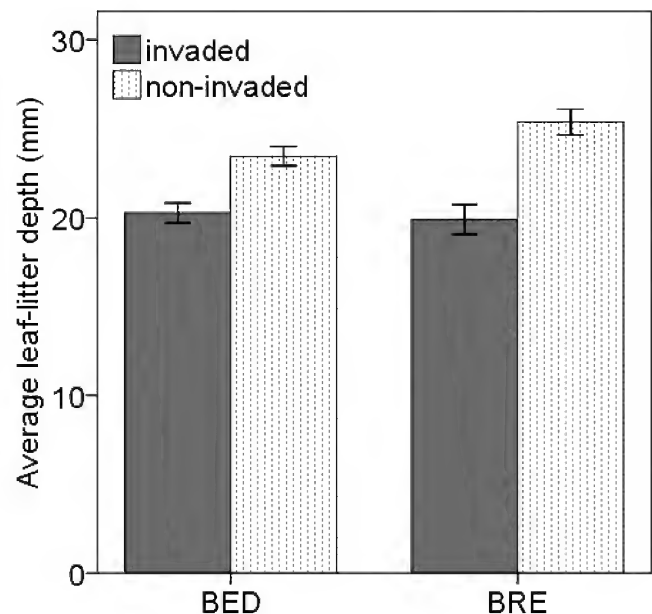


Fig. 6 Average leaf-litter depth at garlic mustard invaded and non-invaded plots. Error bars are $\pm 1\text{SE}$

between the invaded and non-invaded plots.

Although ants have been shown to be good indicators of environmental perturbations and habitat disturbance (Alonso 2000; Kaspari and Majer 2000), the ecological effects of garlic mustard may be too subtle for them to respond, or these effects may be masked by other more potent factors. Site characteristics are important in affecting the outcome of the presence and therefore the impacts of invasive weeds (Samways et al. 1996; Yeates and Williams 2001). The local ant assemblages, at our study sites, likely have been affected by factors related to urbanization and habitat modification prior to the establishment of garlic mustard. Urban forests typically consist of isolated patches with reduced area and high proportion of edges. Anthropogenic habitat alterations such as clearing of native vegetation, fragmentation and destruction of natural habitats and the associated creation of habitat edges operate on large spatial scales and are among the key environmental features of urban landscapes that are known to affect local diversity (Bolger et al. 2000; Gibb and Honchuli 2002). These environmental disturbances are among the major forces shaping the physical and biological changes occurring in urban landscapes (Halme and Niemelä 1993; Murcia 1995; Ewers et al. 2007). Generalist and opportunistic ant species are known to respond more successfully to environmental disturbances than specialist species (Didham et al. 1996; Gibb and Honchuli 2002), and changes in the proportion of opportunistic ant species have been correlated with levels of disturbance (Andersen 1990). Urban forest fragments typically are characterized by a large proportion of generalist species and disturbed-habitat specialists that have adapted to these, often degraded, environments (King and Tschinkel 2006; Clarke et al. 2008). Thus the absence of significant responses from the local ant communities to garlic mustard presence may be a result of the common occurrence, at our study sites, of species capable of tolerating high levels of disturbance. The impacts associated with garlic mustard presence may not be strong enough to elicit further changes in the abundance, richness, and composition of these communities.

Another reason for a lack of detectable differences between the invaded and non-invaded plots may be the relatively recent detection of garlic mustard in the Cleveland area, and in northeast Ohio in general. Although garlic mustard has been present in parts of the US for over a century, it was first detected in 'natural' settings in the Cleveland area in the early 1980s (C. Thomas, pers. communication). This species was likely present earlier in gardens and other human habitations, however it was not a prominent part of the local flora until the late 1980s. Older settlements in Berea, Chagrin Falls, Bedford and Brecksville likely were epicenters for the plant's establishment from which it naturalized urban green spaces (C. Thomas pers. communication).

A few ant species responded to the presence of garlic mustard with changes in their frequency of occurrence and/or abundance but these species were rather uncommon and their responses often were inconsistent across sites. These differences may be a result of patchy distributions and low probability of detection rather than a function of garlic mustard presence. *Formica neogagates* was the only species in our study that showed preferences to areas not invaded by garlic mustard. This woodland species is rare in Ohio, and in the Cleveland region where it is currently known only from Brecksville Reservation (Ivanov unpublished). It is possible that this species is negatively affected by the presence of garlic mustard, but the fact that it only occurred in three samples from a single plot prevents us from making any definitive conclusions about its response to garlic mustard presence. The lack of significant difference in terms of community composition and structure suggest that the presence of garlic mustard did not lead to alterations of local species makeup and community structure.

We were able to document a decrease in the leaf-litter layer at the garlic mustard invaded plots which was correlated with increased abundance of non-native *Amyntas* earthworms. Our results thus are consistent with the findings of Dávalos and Blossey (2004) and Nuzzo et al. (2009). Garlic mustard presence thus may be indicative of underground invasions by non-native *Amyntas* earthworms. These earthworms were more abundant at the garlic

mustard plots, but were also present at a few reference plots. A decrease in the leaf-litter layer has been associated with reduced invertebrate abundance and changes in community structure (Koivula et al. 1999), likely as a direct result of decreased microhabitat and resource heterogeneity. Although we found significant reduction in the leaf-litter layer at the garlic mustard invaded plots, the decrease was small (3.2mm at BED and 5.5mm at BRE) and may not be sufficient to translate into changes in the local forest ant communities. Exotic *Amyntas* have been detected in the region only recently (M. Walton, pers. communication) and it may be that more time is necessary before the impacts associated with their presence are translated into changes in the local leaf-litter arthropod assemblages.

Despite our findings and the similar results reported by Dávalos and Blossey (2004), it is likely that garlic mustard has negative ecological impacts, but the biennial nature of the plant and the pattern of spread may produce subtle effects that are difficult to detect. Moreover, these impacts may be masked by stronger factors such as land use history and habitat disturbance associated with urbanization. Such factors operate at larger spatial scales and may obscure the localized impact of an invasive plant. Future studies focusing on the effects of garlic mustard on local invertebrate communities may benefit by targeting 'undisturbed' natural areas in which garlic mustard is present. Such an approach may help to discriminate between the impacts associated with anthropogenic habitat disturbance and those related to the presence of this invasive species. In addition, the ecological impacts of garlic mustard in temperate deciduous forest may be undetectable when using ants (our study), or ground-dwelling carabid beetles (Dávalos and Blossey 2004) as indicators. The effects of invasive plants on local animal communities have been shown previously to be strongly dependent on the taxonomic group investigated (de Groot et al. 2007). Long-term investigations, experimental manipulations of garlic mustard densities and use of other animal indicators are necessary to more fully reveal the impacts associated with *A. petiolata* invasions (Blossey 1999).

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